Biological Sciences: Ecology

**Title: Biotic homogenization can decrease landscape-scale forest multifunctionality**

**Short title: Beta-diversity and landscape-scale multifunctionality**

AUTHORS: Fons van der Plas1,2\*, Peter Manning1,2, Santiago Soliveres1, Eric Allan1, Michael Scherer-Lorenzen3, Kris Verheyen4, Christian Wirth5,6, Miguel A. Zavala7, Evy Ampoorter4, Lander Baeten4,8, Luc Barbaro9, Jürgen Bauhus10, Raquel Benavides3, Adam Benneter10, Damien Bonal11, Olivier Bouriaud12, Helge Bruelheide6,13, Filippo Bussotti14, Monique Carnol15, Bastien Castagneyrol9, Yohan Charbonnier9, David Coomes16, Andrea Coppi14, Cristina C. Bastias17, Seid Muhie Dawud18, Hans De Wandeler19, Timo Domisch20, Leena Finér20, Arthur Gessler21, André Granier11, Charlotte Grossiord22, Virginie Guyot9,23, Stephan Hättenschwiler24, Hervé Jactel9, Bogdan Jaroszewicz25, François-Xavier Joly24, Tommaso Jucker16, Julia Koricheva26, Harriet Milligan26, Sandra Müller3, Bart Muys19, Diem Nguyen27, Martina Pollastrini14, Sophia Ratcliffe5, Karsten Raulund-Rasmussen18, Federico Selvi14, Jan Stenlid27, Fernando Valladares17,28, Lars Vesterdal18, Dawid Zielínski25, Markus Fischer1,2,29

AUTHOR AFFILIATIONS:

1Institute of Plant Sciences, University of Bern, Altenbergrain 21, 3013 Bern, Switzerland

2Senckenberg Gesellschaft für Naturforschung, Biodiversity and Climate Research Centre, Senckenberganlage 25, 60325 Frankfurt, Germany

3Faculty of Biology, Geobotany, University of Freiburg, Schaenzlestrasse 1, 79104 Freiburg, Germany

4Forest & Nature Lab, Department of Forest and Water Management, Ghent University, Coupure links 653, 9000 Ghent, Belgium

5Systematic Botany and Functional Biodiversity, University of Leipzig, Johannisallee 21, 04103 Leipzig, Germany

6German Centre for Integrative Biodiversity Research (iDiv) Halle-Jena-Leipzig, Deutscher Platz 5e, 04103 Leipzig, Germany

7Forest Ecology and Restoration Group, Department of Life Sciences, University of Alcalá, 28805 Alcalá de Henares, Madrid,  Spain

8Terrestrial Ecology Unit, Department of Biology, Ghent University, K L Ledeganckstraat 35, 9000 Ghent, Belgium

9INRA, UMR 1202 BIOGECO, F-33610 Cestas, France; Univ. Bordeaux, BIOGECO, UMR 1202, F-33600 Pessac, France

10Faculty of Environment and Natural Resources, Chair of Silviculture, University of Freiburg, Fahnenbergplatz, 79085 Freiburg, Germany

11INRA, UMR EEF, 54280 Champenoux, France

12Faculty of Forestry, Stefan cel Mare University of Suceava, Strada Universității 13, Suceava 720229, Romania

13Institute of Biology / Geobotany and Botanical Garden, Martin Luther University Halle Wittenberg, Neuwerk 21, 06108 Halle, Germany

14Department of Agri-Food Production and Environmental Science (DISPAA), Lab. of Applied and Environmental Botany, University of Firenze, Piazzale delle Cascine, 18-50144 Florence, Italy

15Laboratory of Plant and Microbial Ecology, Department of Biology, Ecology, Evolution, University of Liège, Boulevard du Rectorat 27, 4000 Liège, Belgium

16Forest Ecology and Conservation, Department of Plant Sciences, University of Cambridge, Dowwning Street, Cambridge CB2 3EA, United Kingdom

17Department of Biogeography and Global Change, National Museum of Natural Sciences, MNCN, CSIC, Serrano 115 bis, 28006 Madrid, Spain.

18Department of Geosciences and Natural Resource Management, University of Copenhagen, Rolighedsvej 23, 1958 Frederiksberg C, Denmark.

19Division of Forest, Nature and Landscape, University of Leuven, Celestijnenlaan 200E  
 B-3001 Leuven-Heverlee, Belgium

20Natural Resources Institute Finland, P.O. Box 68, FI-80101 Joensuu, Finland

21Swiss Federal Research Institute WSL, Zürcherstr, 111, CH-8903 Birmensdorf, Switzerland

22Earth and Environmental Sciences Division, Los Alamos National Laboratory, P.O. Box 1663   
Los Alamos, NM 87545, LosAlamos, NM, USA

23INRA, UMR 1201, DYNAFOR, F-31326 Castanet-Tolosan, France

24Centre of Evolutionary and Functional Ecology (CEFE UMR 5175 – University of Montpellier – University Paul-Valéry Montpellier – EPHE), 1919 route de Mende, 34293 Montpellier Cedex 5, France

25Białowieża Geobotanical Station, Faculty of Biology, University of Warsaw, Sportowa 19, 17-230 Białowieża, Poland

26School of Biological Sciences, Royal Holloway University of London, Egham, Surrey, TW20 0EX United Kingdom

27[Department of Forest Mycology and Plant Pathology](http://www.slu.se/mykopat), Swedish University of Agricultural Sciences, Box 7026, SE-750 07, Uppsala, Sweden

28Departamento de Biología y Geología, ESCET, Universidad Rey Juan Carlos, c/ Tulipan s.n., 28933 Móstoles, Spain.

29Oeschger Centre for Climate Change Research, University of Bern, Falkenplatz 16  
CH-3012 Bern, Switzerland

CORRESPONDING AUTHOR:

\*Fons van der Plas

Senckenberg Gesellschaft für Naturforschung

Biodiversity and Climate Research Centre

Senckenberganlage 25

60325 Frankfurt

Germany

[fonsvanderplas@gmail.com](mailto:alfons.vanderplas@ips.unibe.ch)

+41 31 631 49 11

KEYWORDS: beta diversity, biodiversity, ecosystem functioning, FunDivEUROPE, spatial scale

ABSTRACT: Many experiments have shown that local biodiversity loss impairs the ability of ecosystems to maintain multiple ecosystem functions at high levels (multifunctionality). In contrast, the role of biodiversity in driving ecosystem multifunctionality at landscape scales remains unresolved. We used a comprehensive pan-European dataset including 16 ecosystem functions measured in 209 forest plots across six European countries and performed simulations to investigate how local plot-scale richness of tree species (α-diversity) and their turnover between plots (β-diversity) are related to landscape-scale multifunctionality. After accounting for variation in environmental conditions, we found that relationships between α-diversity and landscape-scale multifunctionality varied from positive to negative depending on the multifunctionality metric used. In contrast, when significant, relationships between β-diversity and landscape-scale multifunctionality were always positive, as a high spatial turnover in species composition was closely related to a high spatial turnover in functions that were supported at high levels. Our findings have major implications for forest management and indicate that biotic homogenization can have previously unrecognized and negative consequences for large-scale ecosystem multifunctionality.

SIGNIFICANCE STATEMENT

Numerous studies have demonstrated the importance of biodiversity in maintaining multiple ecosystem functions and services (multifunctionality) at local spatial scales, but it is unknown whether similar relationships are found at larger spatial scales in real-world landscapes. Here we show for the first time that biodiversity can also be important for multifunctionality at larger spatial scales in European forest landscapes. Both high local (α-) diversity and a high turnover in species composition between locations (high β-diversity) were found to be potentially important drivers of ecosystem multifunctionality. Our study provides evidence that it is important to conserve the landscape scale biodiversity that is being eroded by biotic homogenization if ecosystem multifunctionality is to be maintained.

\body

It is widely established that high local-scale biodiversity increases levels of individual ecosystem functions in experimental ecosystems (1-4), and that biodiversity is even more important for the simultaneous maintenance of multiple functions at high levels, i.e. ecosystem multifunctionality (5-8). Because the capacity of natural ecosystems to maintain multiple functions and services is crucial for human well-being (9), the positive diversity-multifunctionality relationship is often used as an argument to promote biodiversity conservation (6, 10). However, while society seeks to maximize the delivery of potentially conflicting ecosystem services such as food production, bioenergy generation and carbon storage at the landscape scale (11-13), research into the relationship between biodiversity and ecosystem multifunctionality has been largely limited to local scale studies, where diversity is manipulated in experimental plant communities. While some studies have focused on more natural communities distributed over larger spatial extents (e.g. 14-16), they examined relationships between local-scale biodiversity and local-scale multifunctionality. The only previous study to investigate multifunctionality at larger scales (17) simulated artificial landscapes using data from experimental grassland communities. It showed that while different aspects of biodiversity affected multifunctionality, local scale (α-) diversity was a much stronger driver than the turnover of species between sites (β-diversity). However, whether those findings can be extrapolated to real-world (i.e. natural and semi-natural) ecosystems, such as forests, is unknown. As a result, we have a poor understanding of how multifunctionality relates to biodiversity at the larger spatial scales that are most relevant to ecosystem managers. This is of particular concern given recent findings, which suggest that human driven homogenization of communities (loss of β-diversity (18-21)) may be just as widespread as local (α-) diversity declines (22, 23).

Multifunctionality can be measured by a variety of methods, and the most appropriate means of doing this remains unresolved (24-27), particularly at larger scales, where the desired distribution of ecosystem function across the landscape has not been quantified. At local scales, one can quantify ecosystem multifunctionality as the number of ecosystem functions that exceed a given threshold value, where the threshold equals a certain percentage of the maximum observed value of each function (hereafter ‘threshold-based multifunctionality’; Fig. 1B) (10, 24). This threshold reflects the minimum value of ecosystem functioning that is deemed satisfactory. As trade-offs between ecosystem functions or services are commonplace (5, 7, 28, 29), it is often impossible to maximize all of the desired functions in a local community (6). However, when different species provide different functions (5, 7), at larger spatial scales a high spatial turnover in community composition (i.e. a high β-diversity) across the landscape can cause different parts of the landscape to provide different functions at high levels (defined as high threshold-based β-multifunctionality; Fig. 1B). Therefore, high β-diversity might cause all desired ecosystem functions to be provided at high levels in at least one patch within a landscape (and hence promote *threshold-based* landscape-scale or γ-multifunctionality (Fig. 1B; see also (30)), but only if (a) species differ in the functions they support and (b) there is no ‘superspecies’ that supports the majority of functions. This may be relevant for cases where forest landscapes are managed for many different services (e.g. timber production, the limitation of nutrient runoff and ecotourism), but where each of these only needs to be provided at high levels in a part of the landscape, not everywhere (31). Alternatively, a manager may seek to promote the total delivery of many summed individual ecosystem functions across a landscape. We define this as *sum-based* γ-multifunctionality (Fig. 1B). This may be a more appropriate measure of multifunctionality in cases where the benefits of ecosystem services are manifested at large scales, such as carbon sequestration or water purification (32). In this case β-diversity might only promote sum-based γ-multifunctionality if non-additive diversity effects, such as resource partitioning, species-environment matching or spillover effects operate at relatively large spatial scales (33-34). It is therefore likely that the importance of β-diversity for γ-multifunctionality varies depending on the desired pattern of ecosystem service provision.

Forests provide many ecosystem services including wood production, the regulation of water quality and climate, and recreation (35-36). Most present-day European forests and almost all forest plantations worldwide are dominated by only one or a few tree species (15, 37), although their diversity could be promoted relatively easily by planting more species or by encouraging natural regeneration. This makes the understanding of diversity-multifunctionality relationships in these ecosystems highly relevant for forest management.

We therefore assessed the importance of α- and β-diversity of tree species in driving landscape scale (γ-) multifunctionality in mature European forests. To do this we used data taken from a pan European forest dataset consisting of 209 forest plots, specifically selected to investigate relationships between tree diversity and ecosystem functioning by maximizing variation in dominant ‘target’ species richness while minimizing (a) variation in other potential drivers of ecosystem function (e.g., soil and climatic conditions) and (b) covariation between tree α-diversity, species composition and environmental variables as much as possible (38). Our plot selection therefore aimed to mimic biodiversity experiments in order to investigate relationships between biodiversity and ecosystem functioning in mature forests, which are difficult to undertake with manipulative approaches due to the longevity of tree species. The plots were widely distributed across six European countries, spanning boreal to Mediterranean zones and representing six major European forest types (38). In each plot 16 ecosystem processes, functions or properties (termed ‘functions’ hereafter) were measured. These represented a wide range of supporting, provisioning, regulating and cultural ecosystem services (Table S3; *sensu* (9)). Next, we created simulated landscapes by randomly drawing plots from a country to generate a ‘landscape’ of five plots, from which γ-multifunctionality was calculated. We then explored relationships between α- and β-diversity and different measures of γ-multifunctionality: *threshold-based γ-multifunctionality*, quantified as the number of functions with levels above a threshold (certain percentage of maximum functioning observed across all plots (10)) in at least one plot of the landscape (for quantification, see Fig. 1B), and *sum-based γ-multifunctionality*, quantified as the sum of scaled values of all functions across all plots within a landscape (for quantification, see Fig. 1B). To demonstrate *how* α- and β-diversity can promote threshold-based γ-multifunctionality we also measured the relationships between both α- and β-diversity, and threshold-based α- and β-multifunctionality (for quantification, see Fig. 1B).

RESULTS AND DISCUSSION

Our analyses show that relationships between α-diversity and threshold-based γ-multifunctionality varied from positive, when moderate levels of ecosystem functioning were desired (40-70% thresholds), to negative when very high levels (90% threshold) of ecosystem functioning were required (Fig. 2C; all P < 0.05). In contrast, relationships between β-diversity and threshold-based γ-multifunctionality were, when significant, always positive, irrespective of the level of functioning desired (Fig. 2C, all P < 0.05). These positive relationships with β-diversity were generally consistent throughout countries (Fig. 3C) and largely independent of whether diversity was measured as total species richness or that of abundant target species (Fig S3, Table S5-7) and the statistical approach used to investigate diversity-multifunctionality relationships (Fig. S5-S9). Thus, landscapes with a high spatial turnover in species composition had consistently more functions at high levels in at least some plots than more biotically homogeneous landscapes. This indicates that biotic homogenization can have detrimental consequences for threshold-based landscape-scale (γ-) ecosystem multifunctionality, while management that promotes a higher spatial turnover in species composition may reverse these detrimental effects. In contrast, sum-based γ-multifunctionality was related to neither α- nor β-diversity (Fig. 2C; Fig S3).

Next we investigated the mechanisms by which α- and β-diversity may affect threshold-based γ-multifunctionality by investigating relationships between α- and β-diversity and threshold-based α- and β-multifunctionality (i.e. local multifunctionality and turnover in functioning across plots, Fig. 1B). These analyses showed that the aforementioned relationships between α-diversity and threshold-based γ-multifunctionality were mediated by effects on threshold-based α-multifunctionality: α-diversity was positively related to threshold-based α-multifunctionality when moderate levels (40-50%) of functioning were desired (similar to most experimental studies (8)), but negatively related when high levels (90%) of functioning were required (Fig. 2A; all P < 0.001), a finding that was largely consistent throughout countries (Fig. 3A) and largely independent of whether the richness of the dominant species (results in the main text) or the richness of all species (Fig. S3) were used as local (α-) diversity measures. This pattern may have been caused by ‘statistical averaging’ effects similar to the portfolio effects that drive diversity-stability relationships (39): without strong selection or complementarity effects (40), mixed species plots will tend to have intermediate, but never extremely high or low, ecosystem function levels due to the averaging of individual species effects on function. In line with this, α-diversity did not have significant effects on sum-based γ-multifunctionality (Fig. 2C). These results suggest that while function values were *on average* not higher or lower in diverse communities than in monocultures, they tended to be *less extreme* (never extremely high or low) (41). This result contrasts with other studies focusing on more diverse, experimental grassland, aquatic or soil communities. In these, higher local (α-) diversity enhances α-multifunctionality even at very high thresholds (8, 24), possibly due to strong complementarity effects. However, the diversity-ecosystem functioning literature has tended to concentrate on particular ecosystems and study designs. As a result it is difficult to infer whether these contrasting results are caused by biological or methodological differences. In any case, our results indicate that, in European forests at least, the relationship between local (α-) diversity and both local (α-) and landscape-scale (γ-) multifunctionality strongly depends on the desired level of functioning.

In our next analysis we investigated the relationship between β-diversity and landscape (γ-) multifunctionality. In contrast to α-diversity, the positive relationship between β-diversity and threshold-based landscape (γ-) multifunctionality was independent of the desired level of ecosystem functioning. In almost all countries (Fig. 2B; Fig. 3C) and irrespective of whether target or total species richness was the diversity metric used (Fig. S3), β-diversity was positively related to threshold-based β-multifunctionality when moderate or high levels of ecosystem functioning were desired, thereby increasing the number of functions that were provided at high levels in at least one part of the landscape (threshold-based γ-multifunctionality; Fig. 2B). However, we did not detect a significant relationship between β-diversity and sum-based γ-multifunctionality (Fig. 2B). This was likely due to trade-offs between ecosystem functions: out of the 120 possible pairwise correlations among functions, 50 were negative. This made it impossible to achieve very high levels of all functions across the entire landscape. These results thus indicate that while β-diversity is not related to higher *average* levels of ecosystem functions, it is positively related to the number of functions that perform at high levels in at least part of the landscape. Hence, positive relationships between β-diversity and threshold-based β-multifunctionality (and hence threshold-based γ-multifunctionality) are caused by the fact that different species support different functions (5, 7). For example, in Polish forests, monoculture plots of the conifer *Picea abies* are related to high levels of many functions relating to the production of quality timber (e.g. timber quality, biomass production), while plots of the deciduous tree *Carpinus betulus* were of higher recreational and conservation value due to a high diversity of bats and understorey plants (Table S8). Hence, forest landscapes where some locations were dominated by *Picea abies* and others by *Carpinus betulus* provided more functions at high levels than those where all plots had the same tree species composition.

Our finding that the relationships between biodiversity and European landscape-scale (γ-) forest multifunctionality depends strongly on the way that multifunctionality is quantified, has important implications for European forest management. In short, our results suggest that different patterns of tree species distribution would achieve different management goals (or ‘landscape multifunctionality scenarios’). The results of the threshold-based γ-multifunctionality analysis would be most relevant to situations where managers sought to promote forest landscapes with very high levels of ecosystem functioning in at least some (but not necessarily all) local patches (sensu 30). As described earlier, this may occur when managers seek to provide different ecosystem services in different localities. An example of such a landscape is one where some localities provide recreation or cultural services, such as aesthetic beauty and a diversity of charismatic taxa, (see Supplementary Material S3) (31), while other localities maximize provisioning services that are only cost-effective when delivered at very high levels, e.g. the production of high value timbers (42), or form hotspots of certain biogeochemical functions that need to be strategically located, such as the minimization of nutrient runoff close to water bodies. In such scenarios, threshold-based γ-multifunctionality could be promoted in forest landscapes that possess a high turnover in community composition, but a *low* local (α-) diversity (top left landscape in Fig. 1C), i.e. by promoting a range of different monocultures across the landscape. When the delivery of provisioning services is cost-effective at lower levels, or when cultural or regulating services do not need to be at extremely high levels, one could aim to promote landscapes with moderate levels of many functions (40 or 50% threshold γ-multifunctionality). In this scenario, threshold-based γ-multifunctionality is highest in forests with both a high spatial turnover in community composition *and* a high local diversity of tree species (top right landscape in Fig. 1C). This is in line with Gamfeldt *et al.* (15) who hypothesized, based on local-scale analyses, that “adjacent stands, each with multiple species but in different combinations, might be the best way to provide multiple ecosystem services at the landscape scale”. A third hypothetical scenario would be to maximize total delivery of services across the landscape (high sum-based γ-multifunctionality), rather than having highly localized specialist patches that deliver a limited number of services at very high levels. This scenario may be most relevant to cases where the primary goal of ecosystem management is to provide ecosystem services whose benefits are manifested at large-scales, such as carbon sequestration (43). We found that neither local scale, α-diversity nor spatial turnover in community composition (β-diversity) had significant detectable relationships with γ-multifunctionality under this scenario. In summary, we demonstrate that the importance of different components of diversity for promoting landscape-scale (γ-) multifunctionality is likely to depend on management goals. Accordingly, stakeholder engagement is required to see where these situations apply in real forested landscapes. Further studies are also required to confirm that tree α- and β-diversity are causal drivers of the observed relationships and to see how important they are in comparison to other potentially important factors in driving ecosystem multifunctionality in representative, European forests.

In this study, some of the benefits of biodiversity for landscape-scale (γ-) multifunctionality may have been underestimated. Our study did not consider some of the spatiotemporal processes that occur in real forest landscapes, e.g. the dispersal and movement of ecosystem service providers, species-environment matching, large-scale resource partitioning or the spill-over and subsidy of ecosystem services between neighboring patches (13, 44). These processes could promote ecosystem functioning in landscapes that possess a high spatial turnover in species composition even more than was detected here. For example, a forest resistant to herbivory might also reduce pest damage in adjacent forests by lowering populations of herbivores and by preventing their movement into more vulnerable areas, thus strengthening the relationship between β-diversity and landscape-scale (γ-) forest multifunctionality. Future studies could explore these ideas by studying ecosystem multifunctionality in landscapes where ecological interactions between patches of differing diversity and composition are quantified.

Previous studies have demonstrated that local scale (α-) biodiversity can boost multifunctionality in the real world, in addition to experimental, ecosystems (8, 14-16). Here, we add evidence that both local-scale (α-) and β-diversity can also drive ecosystem multifunctionality at the landscape scale, and that the desired distribution of ecosystem function across the landscape influences the importance of this relationship. Biotic homogenization is occurring worldwide at local, regional and global scales (19-21). Similarly, current forest management often results in large areas of low species turnover. Our study is an important step forward in exploring the importance of this biotic homogenization for landscape-scale (γ-) multifunctionality. It shows that biotic homogenization may have negative, strong, far reaching and so far overlooked impacts upon the ecosystem services on which humanity depends, and that these may be as strong, or even stronger than those of local diversity loss.

METHODS

Plot selection

In total, 209 30 x 30 meter forest plots were established within the European FunDivEUROPE project (<http://fundiveuropektp.boku.ac.at/>). Because we were interested in the effects of tree species diversity on ecosystem functioning in mature forests (38), plot selection was aimed to mimic the design of a biodiversity experiment, in which variation in environment is minimized and diversity is not confounded with composition, as in most observational studies of diversity. Hence, the design aimed to bridge the gap between controlled but very young tree diversity experiments and observational studies where diversity can be strongly confounded with other factors.

Plots were located in six European countries, ranging from boreal to Mediterranean zones, and with each representing a major European forest type (38): Finland (28 plots; boreal forest), Poland (43 plots; hemiboreal forest), Germany (38 plots; temperate deciduous forest), Romania (28 plots; mountainous deciduous forest), Italy (36 plots; thermophilous deciduous forest), and Spain (36 plots; Mediterranean mixed forest) (Fig. S1). Within countries, plots were located in a single region ranging in size from 5 x 5 km (Romania) to 150 x 150 km (Finland). In each country between 3 and 5 regionally common "target" species were selected, 15 in total (Table S1). Plots were then selected to differ as much as possible in richness of target species and so that almost all possible combinations of these target species were realized, a design that emulates those of biodiversity experiments (38). Richness levels of 1, 2, 3, 4 and 5 target species were respectively replicated 56, 67, 54, 29 and 3 times, across countries, and most possible target species compositions were realized (for further details on the selection procedure, see (38)). To achieve this goal, some admixture of non-target species was unavoidable. However, target species on average accounted for 93.75% of the individuals and 91.39% of the basal area and they were always represented by >2 individuals (see Figs S2). We therefore focus on using the richness of target species in our analyses but also tested for the effect of total (target + admixed) species richness (Supplementary Material). Plot selection strictly avoided correlations between tree species richness and soil factors, as well as any spatial autocorrelation in diversity (38) by choosing plots that differed as little as possible in environmental factors (soil texture, depth and pH, and altitude) that could potentially confound diversity effects on multifunctionality. The diversity gradient was therefore most likely a result of stochastic factors or differences in past management between plots.

Tree diversity and community composition data

Within each plot, all tree stems ≥7.5 cm in diameter at breast height were identified to species and mapped (12,939 stems in total). Species richness was defined as the number of target species (Table S1) with at least 2 individuals in a plot. We also calculated Pielou’s evenness (45) for target tree species and the proportion of coniferous target trees. Because plots were specifically selected to have similar abundances of the target species, variation in evenness values across plots was low, with values above 0.6 in >90% of plots. In addition to richness, evenness and the proportion of coniferous individuals of *target* species, we also calculated richness, evenness and proportion of coniferous individuals of *all* tree species for the purpose of sensitivity analyses.

We recorded diameter to the nearest 0.1cm of each individual tree stem and measured height to the nearest 0.1m. We used these diameter and height measurements to estimate aboveground biomass of each tree individual, based on published allometric functions (see (46) and references therein). These functions were species-specific, and whenever possible, functions were developed for trees growing in forests similar to those of our study were used. Plot-level biomass estimates were calculated by summing the biomass of all individuals of the target tree species within a plot.

Environmental data

We recorded the altitude of each plot as a proxy for variation in local climate. Soil pH was also measured because it is an important driver of numerous other soil properties (47). Between May and October 2012, forest floor litter (in nine 25x25 cm patches) and mineral soil (using a cylindrical metal corer to 0-10 in all countries and 10-20 cm in all countries but Spain) were sampled for pH measurements, which were then measured using standard protocols (see Supplement). Soil texture was also estimated using expert assessment as the abundance of sand (size), silt (size) and clay (size) content. Measurements were done on an ordinal scale, with values ranging from 1 (absent) to 3 (very common). Finally, soil depth (cm depth to bedrock) was measured in each plot using a soil auger.

Measurement of ecosystem functions and properties

In each plot, 16 ecosystem processes, functions or properties (termed ‘functions’ hereafter) were measured between 2012 and 2014: timber quality, timber production, tree regeneration, root biomass, litter decomposition, wood decomposition, microbial biomass, soil carbon stock, resistance to drought, resistance to insect herbivory, resistance to mammal browsing, resistance to pathogens, bird diversity, bat diversity, understory plant diversity and earthworm biomass. All measured ecosystem functions have established links to supporting, provisioning, regulating or cultural services (*sensu* (9)). For details about function measurements, see Supplementary Material S2 and for details on the services they provide, see Supplementary Material S3. To allow comparison of the different ecosystem functions they were scaled between 0 and 1: , with *SEF* indicating the final (scaled) ecosystem value, *EF* indicating raw (unscaled) ecosystem function values and *min/max(EF)* respectively indicating the minimum/maximum raw values of the ecosystem function.

Simulating artificial forest landscapes

To analyze diversity and multifunctionality at different spatial scales, ranging from plots (α), species turnover between plots (β), to landscape scales (γ), we simulated artificial forest landscapes from the observed forest plots. Within each country, we randomly selected, without replacement, five plots to create an artificial landscape and repeated this process 1000 times. With six countries, we therefore created 6000 artificial forest landscapes, with 5981 unique plot combinations. The number of unique dominant tree species within countries was relatively small (up to five) and few plots contained all of these, hence creating landscapes from a relatively low number of plots ensured that landscapes varied as much as possible in both α- and β-diversity. Additional analyses showed that the compositions of these simulated landscapes are likely to be realized at the local regional scale (Supplementary Material S5). Within each of these 5981 unique landscapes, we then calculated tree diversity at two spatial scales. Local (α-) scale diversity was defined as the average target species richness value across the plots. Turnover in tree community composition (i.e. ß-diversity; (48)) was calculated for each of the ten pairs of plots within a landscape as: , where A and B are the number of target species unique to each plot and C is the number of target species shared by the plots (49). This measure is bound between 0 (no turnover) and 1 (complete turnover). Landscape-level ß-diversity was then calculated as the average of all ten ß-diversity values of pairwise plot combinations. γ-diversity was calculated as the richness of all the target species present in at least one plot within the landscape. Note that γ-diversity (or threshold-based γ-multifunctionality, see below) is not strictly additively or multiplicatively partitioned into α- and β-diversity (or threshold-based α- or β-multifunctionality, see below), so that α-, β- and γ-diversity can to some extent vary independently from one another. For sensitivity analyses, we also calculated α-, β- and γ-diversity based on all tree species present (rather than target species only).

In the 5981 artificial landscapes, we used two approaches to calculate multifunctionality measures, which correspond to different hypothetical management objectives. We calculated threshold-based α-, ß- and γ-multifunctionality in a way that is analogous to calculating α-, ß- and γ-diversity (Fig. 1) and also broadly analogous to a recent method for quantifying the temporal stability of ecosystem functioning at different spatial scales (50). Within plots, *threshold*-based multifunctionality was defined as the number of ecosystem function values that exceeded a minimum threshold: (10), in which *n* is the number of functions and *T* is the performance threshold value. Threshold values were defined as a certain percentage of the 95th percentile of maximum functioning (10) from the country in which plots were located. We chose to investigate diversity-multifunctionality relationships at four different multifunctionality thresholds: 40, 50, 70 and 90%. In plots with one (n=28), two (n=1) or three (n=1) missing ecosystem function values, threshold-based multifunctionality scores were corrected by accounting for the proportion of non-missing functions: , where *TMF* is threshold-based multifunctionality, *n\_c* is the number of non-missing functions and *n* is the total number of functions measured in this study. Threshold-based α-multifunctionality was then calculated as the average *TMF* value across the five plots comprising a landscape. Threshold-based β-multifunctionality was calculated as the turnover in ecosystem functions present (i.e. exceeding a threshold) across plots comprising a landscape, with the same formula as was used for ß-diversity (see above), but this time with A and B representing functions exceeding the threshold in *either* the first or the second plot and C representing the ecosystem functions that exceed the threshold in *both* plots. Lastly, threshold-based γ-multifunctionality was measured as the number of ecosystem functions exceeding either a 40, 50, 70 or 90% threshold value in at least one of the five plots within each landscape.

In addition to the threshold based approach, we also calculated γ-multifunctionality based on a summing approach (broadly similar to that used by (14)). To calculate sum-based γ-multifunctionality, we first summed the five plot-level values for each function. These summed landscape-level function values were then scaled between 0 and 1 (). To calculate sum-based γ-multifunctionality we then summed the scaled values of the 16 functions. In contrast to biodiversity and threshold-based multifunctionality, sum-based multifunctionality is quantified using continuous variation in function values, rather than on ‘presence/absence data’ (of species or of functions passing a threshold), making it impossible to partition it into α- and β-components.

In the landscapes we also quantified factors that potentially affect relationships between diversity and multifunctionality: average values of target species evenness, the proportion of coniferous tree individuals, sand content, clay content, soil depth, soil pH and altitude. In addition, we calculated environmental heterogeneity in two steps. First we quantified the heterogeneity of individual abiotic factors (altitude, pH and soil texture), as the coefficient of variation (CV) of values across plots within a landscape. In the case of soil texture, heterogeneity was quantified as the sum of CV values of clay, silt and sand content. Next, these three heterogeneity measures were Z-transformed and summed to produce a single measure of environmental heterogeneity. By using Z-scores, we ensured that each abiotic variable had equal impact on total environmental heterogeneity. All analyses were done with R version 3.0.2 (48).

Statistical analyses

We first investigated whether α- and β-diversity in simulated forest landscapes were associated with γ-multifunctionality using Linear Mixed Models (LMMs). While we designed our study to minimize variation in environmental factors, completely eliminating any variation in these was impossible (Supplementary Material S3). Hence, to avoid the detection of spurious diversity-multifunctionality relationships, we included these environmental factors as covariates in the LMMs. We performed three different LMM analyses. In the first, we investigated how α-multifunctionality was driven by α-diversity, using an LMM with α-multifunctionality as the response variable, species richness as the focal fixed factor, species evenness, proportion of evergreen trees, altitude, soil depth, soil pH, soil sand and clay content and their two-way interactions as covariates and with country as a random factor. In the second analysis, we investigated the relationship between β-multifunctionality and β-diversity, using an LMM with β-multifunctionality as the response variable, β-diversity as the focal fixed factor, environmental heterogeneity as a covariate and country as a random factor. In the third analysis, we investigated how γ-multifunctionality was affected by both α- and β-diversity, by first constructing a full LMM with γ -multifunctionality as the response variable, α- and β-diversity as the focal fixed factors, species evenness, proportion of evergreen trees, altitude, soil depth, soil pH, soil sand and clay content, environmental heterogeneity and their bivariate interactions as covariates and with country as a random factor. As a result of the careful study design, diversity measures were largely independent of the covariates (38): correlations between focal predictors and covariates were always <0.230, while the correlation between α- and β-diversity was moderate (R2 = 0.316; P < 0.0001), hence there was no strong indication of multicollinearity. In all three analyses we used a backward model-selection analysis to sequentially remove covariates (based on ratio-likelihood tests with a Bonferroni correction) until we reached a final model with only the focal fixed factor(s) and significant covariates. From this final model we quantified the significance of α- and β-diversity in driving multifunctionality using likelihood ratio tests, and we also quantified their standardized regression coefficients. All analyses were performed for all different threshold-based and sum-based multifunctionality variables. As a robustness check, we also repeated all these analyses with predictors and covariates based on *all* tree species, rather than on target species only (see Supplementary Material). In addition, to investigate how general the main patterns were across countries, we also ran linear models (LMs) for each country separately, with the same fixed factors in models as in the finally selected LMMs. All analyses were performed using R version 3.0.2 (51). LMMs were fitted using the ‘lmer’ function of the ‘lme4’ library (52). Given that effects of covariates were variable and complex and that the main focus of this study was on biodiversity effects, the effects of covariates are not presented here. For effect sizes of all covariates, see Tables S2-4).

In addition to LMMs, we also used Structural Equation Models (SEM) to investigate relationships between biodiversity and multifunctionality, in order to test the sensitivity of our results to the statistical method used, and to test for indirect relationships between biodiversity and multifunctionality (see Supplementary Material S4).

ACKNOWLEDGEMENTS

The research leading to these results received funding from the European Union Seventh Framework Programme (FP7/2007-2013) under grant agreement no. 265171. We also thank Hainich National Park administration as well as Felix Berthold and Carsten Beinhoff for support of this study and Gerald Kaendler and the Johann Heinrich von Thünen-Institut for providing access to the German National Forest Inventory data. Icons in Fig. 1 and S5.1 are licensed as Creative Commons – Attribution (CC BY 3.0) and were developed by paxandassociates, Andrew McKinley, Agne Alesiute and James Keunig for the Noun project <https://thenounproject.com>.

REFERENCES

1. Hooper DU *et al.* (2005). Effects of biodiversity on ecosystem functioning: a consensus of current knowledge. *Ecol. Monogr* 75:3-35.
2. Balvanera P *et al.* (2006) Quantifying the evidence for biodiversity effects on ecosystem functioning and services. *Ecol. Lett.* 9:1146-1156.
3. Cardinale BJ *et al.* (2011) The functional role of producer diversity in ecosystems. *Am. J. Bot.* 98:572-592.
4. Handa IT *et al.* (2014) Consequences of biodiversity loss for litter decomposition across biomes. *Nature* 509:218-221.
5. Hector A, Bagchi R. (2007) Biodiversity and ecosystem multifunctionality. *Nature* 448:188-191.
6. Zavaleta ES, Pasari JR, Hulvey KB, Tilman GD. (2010) Sustaining multiple ecosystem functions in grassland communities requires higher biodiversity. *P. Natl. Acad. Sci. USA* 107:1443-1446.
7. Isbell F *et al.* (2011). High plant diversity is needed to maintain ecosystem services. *Nature* 477:199-202.
8. Lefcheck JS *et al.* (2015) Biodiversity enhances ecosystem multifunctionality across trophic levels and habitats. *Nat. Comm.* 6:6936.
9. Millennium Ecosystem Assessment (2005). Ecosystems and human well-being: synthesis. Island Press.
10. Gamfeldt L, Hillebrand H, Jonsson, PR (2008) Multiple functions increase the importance of biodiversity for overall ecosystem functioning. *Ecology* 89:1223-1231.
11. Green RE, Cornell SJ, Scharlemann JPW & Balmford A. (2005) Farming and the fate of wild nature. *Science* 307:550-555.
12. Goldstein JH *et al.* (2012) Integrating ecosystem-service trade-offs into land-use decisions. *P. Natl. Acad. Sci. USA* 109:7565-7570.
13. Manning P, Taylor G, Hanley ME. (2015) Bioenergy, food production and biodiversity – an unlikely alliance? *GCB Bioenergy* 7:570-576.
14. Maestre FT *et al.* (2012) Plant species richness and ecosystem multifunctionality in global drylands. *Science* 335:214-218.
15. Gamfeldt L *et al.* (2013) Higher levels of multiple ecosystem services are found in forests with more tree species. *Nat. Commun.* 4:art. 1340.
16. Allan E *et al.* (2015) Land use intensification alters ecosystem multifunctionality via loss of biodiversity and changes to functional composition. *Ecol. Lett.* 18:834-843.
17. Pasari JR, Levi T, Zavaleta ES, Tilman D. (2013) Several scales of biodiversity affect ecosystem multifunctionality. *P. Natl. Acad. Sci. USA* 110:10219-10222.
18. McKinney ML, Lockwood JL. (1999) Biotic homogeneization: a few winners replacing many losers in the next mass extinction. *Trends Ecol. Evol.* 14:450-453.
19. Dornelas M *et al.* (2014) Assemblage time series reveal biodiversity change not systematic loss. *Science* 344:296-299.
20. Vellend M. *et al.* (2013). Global meta-analysis reveals no net change in local-scale plot biodiversity over time. *P. Natl. Acad. Sci. USA* 110:19456-19459.
21. McGill BJ, Dornelas M, Gotelli NJ, Magurran AE. (2015) Fifteen forms of biodiversity trend in the Anthropocene. *Trends Ecol. Evol.* 30:104-113.
22. Murphy GEP, Romanuk TN. (2014) A meta-analysis of declines in local richness from human disturbances. *Ecol. Evol.* 4:91-103.
23. Newbold T *et al.* (2015) Global effects of land use on local terrestrial biodiversity. *Nature* 520:45-50.
24. Byrnes JEK *et al.* (2014) Investigating the relationship between biodiversity and ecosystem multifunctionality: challenges and solutions. *Methods Ecol. Evol.* 5:111-124.
25. Bradford MA *et al.* (2014) Discontinuity in the responses of ecosystem processes and multifunctionality to altered toil community composition. *P. Natl. Acad. Sci. USA* 111:14478-14483.
26. Byrnes, JEK *et al.* (2014) Multifunctionality does not imply that all functions are positively correlated. *P. Natl. Acad. Sci. USA* 111:E5490.
27. Bradford MA *et al.* (2014) Reply to Byrnes et al.: Aggregation can obscure understanding of ecosystem multifunctionality. *P. Natl. Acad. Sci. USA* 111:E5491.
28. Lavorel S *et al.* (2011) Using plant functional traits to understand the landscape distribution of multiple ecosystem services. *J. Ecol.* 99:135-147.
29. Grigulis K *et al.* (2013) Relative contributions of plant traits and soil microbial properties to mountain grassland ecosystem services. *J. Ecol.* 101:47-57.
30. Brandt J. (2003) Multifunctional landscapes – perspectives for the future. *J. Env. Sci.* 15:187-192.
31. Chan KMA, Shaw MR, Cameron DR, Underwood EC, Daily G. (2006) Conservation planning for ecosystem services. *PLoS Biol.* 4:e379.

Trumper K *et al.* (2009) The Natural Fix? The role of ecosystems in climate mitigation. A UNEP rapid response assessment. United Nations Environment Programme, UNEPWCMC, Cambridge, UK

1. Loreau M, Mouquet N, Gonzalez A. (2003) Biodiversity as spatial insurance in heterogeneous landscapes. *P. Natl. Acad. Sci. USA* 100:12765-12770.
2. Cardinale BJ, Ives AR, Inchausti P (2004) Effects of species diversity on the primary productivity of ecosystems: extending our spatial and temporal scales of inference. *Oikos* 104:437-450.
3. Aerts R, Honnay O (2011) Forest restoration, biodiversity and ecosystem functioning. *BMC Ecol.* 11:19.
4. Fares S, Mugnozza GD, Corona P, Palahí M. (2015) Five steps for managing Europe’s forests. *Nature* 519:407-409.
5. Bauhus J, van der Meer P, Kanninen M (2010) *Ecosystem goods and services from plantation forests.* Earthscan, London, Washington DC.
6. Baeten L. *et al.* (2013) A novel comparative research platform designed to determine the functional significance of tree species diversity in European forests. *Perspect. Plant Ecol. Evol. Syst.* 15:281-291.
7. Tilman D, Lehman CL, Bristow CE. (1998) Diversity-stability relationships: statistical inevitability or ecological consequence? *Am. Nat.* 151:277–282.
8. Loreau M, Hector A. (2001) Partitioning selection and complementarity in biodiversity experiments. *Nature* 412:72-76.
9. Loreau M. (1998) Biodiversity and ecosystem functioning: A mechanistic model. *P. Natl. Acad. Sci. USA* 95:5632-5636.
10. Phalan B, Onial M, Balmford A, Green RE. (2011) Reconciling food production and biodiversity conservation: land sharing and land sparing compared. *Science* 333:1289-1291.
11. Canadell JG, Raupach MR (2008) Managing forests for climate change mitigation. *Science* 320:1456-1457.
12. Tscharntke T, Klein AM, Kruess A, Steffan-Dewenter I, Thies C. (2005) Landscape perspectives on agricultural intensification and biodiversity – ecosystem service management. *Ecol. Lett.* 8:857-874.
13. Pielou EC. (1966) The measurement of diversity in different types of biological collections. *J. Theo. Biol.* 13:131-144.
14. Jucker T, Bouriaud O, Avacaritei D, Coomes, DA. (2014) Stabilizing effects of diversity on aboveground wood production in forest ecosystems: linking patterns and processes. *Ecol. Lett.* 17:1560-1569.
15. White RE. (2005) Principles and practice of soil science: the soil as a natural resource, 4th editition. Wiley-Blackwell.
16. Whittaker RH. (1960) Vegetation of the Siskiyou Mountains, Oregon and California. *Ecol. Monogr.* 30:279-338.
17. Lennon JJ, Koleff P, Greenwood JJD, Gaston KJ. (2001) The geographical structure of British bird distributions: diversity, spatial turnover and scale. *J. Anim. Ecol.* 70:966-979.
18. Wang S, Loreau M. (2014) Ecosystem stability in space: α, β and γ variability. *Ecol. Lett.* 17:891-901.
19. R Core Team. (2013) R: A language and environment for statistical computing. - R Foundation for Statistical Computing.
20. Bates D, Maechler M, Bolker B, Walker S (2014) *lme4: Linear mixed-effects models using Eigen and S4.* R package version 1.1-7, <https://CRAN.R-project.org/package=lme4>.
21. Jucker T *et al.* (2014) Competition for light and water play contrasting roles in driving diversity-productivity relationships in Iberian forests. *J. Ecol.* 5:1202-1213.
22. Viro PJ. (1952) Kivisyyden määrittämisestä. Summary: On the determination of stoniness. *Comm. Inst. Forest Fenn.* 40:23 p.
23. Vance ED, Brookes PC, Jenkinson DS. (1987) An extraction method for measuring soil microbial biomass C. *Soil Biol. Biochem.* 19:703-707.
24. Skjemstad JO, Baldock JA. (2007) Total and organic carbon. *Soil sampling and methods of analysis, pp225-238*. 2nd ed. CRC Press, Boca Raton, FL.
25. Matejovic I. (1993) Determination of carbon, hydrogen, and nitrogen in soils by automated elemental analysis (dry combustion method). *Commun. Soil Sci. Plant Anal.* 24:2213-2222.
26. Vesterdal L, Schmidt IK, Callesen I, Nilsson LO, Gundersen P. (2008) Carbon and nitrogen in forest floor and mineral soil under six common European tree species. *Forest Ecol. Manag.* 255:35-48.
27. Grossiord C. *et al.* (2014) Tree diversity does not always improve resistance of forest ecosystems to drought. *P. Natl. Acad. Sci. USA* 111:14812-14815.
28. Eichhorn J *et al.* (2010) Visual assessment of crown condition and damaging agents. In UNECE (Ed.), *Manual on methods and criteria for harmonized sampling, assessment, monitoring and analysis of the effects of air pollution on forests* (pp. 49). Hamburg, Germany: UNECE, ICP Forests Programme Co-ordinating Centre.
29. Grace JB. (2006) *Structural Equation Modeling and natural systems.* Cambridge University Press, Cambridge, United Kingdom.
30. Davidar P *et al.* (2007) The effects of climatic gradients, topographic variation and species traits on the beta diversity of rain forest trees. *Global Ecol. and Biogeogr.* 16:510-518.
31. Rhoads BL. (1991) Multicollinearity and parameter estimation in simultaneous-equation models in fluvial systems. *Geogr. Anal.* 23:346-361.
32. Polley H, Schmitz F, Hennig P, Kroiher F. (2010) Germany. In: Tomppo E, Gschwantner T, Lawrence M, McRoberts RE. (2010) National forest Inventory Pathways Common Report. Springer, pp. 223-243.
33. Nakagawa S, Schielzeth H. (2013) A general and simple method for obtaining R2 from generalized linear mixed effects models*. Methods Ecol. Evol.* 4:133-14.

FIGURE LEGENDS

Figure 1. Quantifying biodiversity and multifunctionality across spatial scales. The light yellow areas represent hypothetical landscapes, consisting of (white) local communities. In these communities, some species are present (colored icons in A/C), while others are absent (grey icons). Similarly, some functions are performing above a hypothetical threshold of 0.5 (colored icons in B), while others are not (grey icons). Diversity and threshold-based multifunctionality are quantified at: (i) the local plot (α-) scale as the number of species present (2 and 3 in A) or functions performing above a given threshold (2 and 3 in B); (ii) the ß-scale: the turnover in species composition ( in A (49)) or functions ( in B (49)) across plots, and (iii) the landscape (γ-) scale, as the number of functions (4 in B) present in at least one plot. Sum-based γ-multifunctionality is defined as the sum of all standardized ecosystem values in a landscape (= 0.8 + 0.2 + 0.7 + 0.4 + 0.9 + 1.0 + 0.1 + 0.6 = 4.7). In contrast to threshold-based multifunctionality, sum-based multifunctionality is not analogous to biodiversity (where species are either present or absent), and can therefore not be partitioned into α- or ß-components. This framework allows investigation of whether γ-multifunctionality is promoted by α- and/or ß-diversity (C).

Figure 2. Scale dependent effects of biodiversity on forest ecosystem multifunctionality. Bars represent the standardized regression coefficients of α-diversity (light grey) and β-diversity (dark grey) in GLMMs explaining α- (A), β- (B) or γ- (C) multifunctionality. Multifunctionality was quantified at different scales using a threshold approach, with thresholds of 40, 50, 70 and 90%. In addition, sum-based γ-multifunctionality (‘SMF’) was calculated as the sum of scaled (between 0 and 1) individual function values. Diversity measures were calculated based on individuals of ‘target’ tree species.

Figure 3. Scale dependent effects of biodiversity on forest ecosystem multifunctionality are generally consistent across countries. Bars represent the standardized regression coefficients of α-diversity (blue) and β-diversity (red) in LMMs explaining α- (A), β- (B) or γ- (C) multifunctionality. Multifunctionality was quantified at different scales using a threshold approach, with thresholds of 40, 50, 70 and 90%. In addition, sum-based γ-multifunctionality (‘SMF’) was calculated as the sum of scaled (between 0 and 1) individual function values. Diversity measures were calculated based on individuals of ‘target’ tree species.